Journal of Cleaner Production 171 (2018) 1396–1406

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Climate change mitigation potential of contaminated land redevelopment: A city-level assessment method

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ARTICLE INFO

Article history:
Received 29 April 2017
Received in revised form
7 September 2017
Accepted 8 October 2017
Available online 9 October 2017

Keywords:
Brownfield development
Remediation
Consequential life cycle assessment
Greenhouse gas emission
Climate change mitigation

ABSTRACT

Under the Paris Agreement, nations and cities are under increasing pressure to reduce greenhouse gas (GHG) emission. Brownfield land redevelopment promises climate change mitigation benefits; however, no studies have attempted to quantify the mitigation potential on a city level. Here we propose a method for this purpose. The method is based on life cycle assessment of GHG emission as compared to greenfield land development, in three categories: (1) primary impact (associated with physical state of brownfield sites and greenfield sites), (2) secondary impact (associated with remediation activities at brownfield sites), and (3) tertiary impact (associated with post-remediation usage of the brownfield sites and avoided usage of greenfield land). A quantitative assessment was conducted for the city of San Francisco in the US, a growing city with ambitious carbon reduction plans. City wide contaminated land remediation activities were found to render a significant secondary impact, resulting in increased GHG emission of 3.54 Mt CO_2 eq. However, by averting the development of greenfield land, this negative impact was partially off-set by a primary impact reduction of 1.22 Mt CO_2 eq. Moreover, the redevelopment of brownfield land offers large tertiary impact reductions due to higher housing density, less utility and road construction needs, shorter commute distances, and smaller household energy consumption. Overall, the results show that brownfield land redevelopment led to a net GHG reduction of 51.9 Mt CO_2 eq. over a 70-year period, or 0.74 Mt CO_2 yr^{-1}, the equivalent of 14% of San Francisco's 5.3 Mt CO_2 eq. GHG emission in 2010. It was found that reduced household electricity and space heating energy consumption and shorter commute distances were the main contributors, accounting for 41.2% and 34.0% of the tertiary impact reduction, respectively. While it is not expected that all cities can fully capture these climate change mitigation benefits, the present study demonstrates a quantification method and emphasizes the large climate change mitigation potential associated with brownfield land redevelopment. Progression of this premise will require regulators and policy makers to better align cities' climate change mitigation strategies with urban planning and development actions.

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1. Introduction

Global climate change represents a serious threat to people's wellbeing in modern society (Intergovernmental Panel on Climate Change, 2014), and it requires multi-level mitigation efforts (Marquardt, 2017; Paul et al., 2017). The latest concerted action by the international community on combating climate change was the signing of the Paris Agreement, which entered into force on November 4, 2016 (UNFCC, 2017). Under the Paris Agreement, each country is bound to ambitious greenhouse gas emission reduction goals, which should become increasingly stringent. While the newly elected President threatened to withdraw the United States (US) from the groundbreaking Paris Agreement, many governmental bodies below the Federal level, including cities, are expected to continue their effort in fighting against global climate change.

Cities are in a position to make a difference in combating climate change. Globally, cities account for 80% of energy use and 70% of

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https://doi.org/10.1016/j.jclepro.2017.10.071
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greenhouse gas (GHG) emissions (U.N. Habitat, 2011). Forty of the globe's largest cities in the C40 network, representing 8% of the global population and 20% of global GDP, announced at the 2012 Earth Summit held in Rio de Janeiro, Brazil, that they have policies in place to reduce GHG emission by 1 billion tons of CO_2 eq. by 2030 (SFDE, 2013). The development of vacant and underdeveloped urban land promises to help cities to reach their climate change mitigation goals. However, much of this land has been contaminated by historical industrial and/or commercial activities. Under the current paradigm of site investigation and remediation, the remediation and redevelopment of contaminated land can take decades; e.g., on average, pump and treat systems in the US operate for 18 years (USEPA, 2004). A newly emerging movement, called "green and sustainable remediation" (GSR) has drawn much attention from both governments and industrial practitioners. The GSR movement has led to research and practice of using more energy efficient technologies and more environmental materials for remediation (Li et al., 2017; Shen et al., 2017a; Wang et al., 2017; Zhang et al., 2017). GSR holds promise in shifting the traditional paradigm to a more holistic decision making processes and to accelerate the redevelopment of contaminated land (Ellis and Hadley, 2009; Hou and Al-Tabbaa, 2014; Prior, 2016).

In the 1980s and 1990s, in the context of decision making for the remediation of contaminated land, academic researchers and government agencies focused on risk management and the removal of toxic contaminants from the original site (termed "primary impact"). In late 1990s and early 2000s, the remediation movement switched attention to life cycle thinking and cradle-to-grave impact of remediation operations (termed "secondary impact") (CLARINET, 2002; Page et al., 1999). It is not until the later 2000s, when researchers first brought up the concept of tertiary impact, which refers to the impacts from redevelopment of contaminated land (Lesage et al., 2007c). To quantify the climate change mitigation potential of brownfield redevelopment, it is considered imperative to account for all three types of impacts, namely GHG emissions 1) associated with the physical state of a brownfield site, 2) due to remediation activities, and 3) reduction as a consequence of brownfield usage.

The present study hypothesizes that on a city-level, it is important to account for tertiary impacts of GHG emissions and mitigation for decision making and policy development. The objectives of this paper include: 1) present a methodology for city-level assessment of the GHG reduction potential by brownfield redevelopment; 2) to quantify and compare primary, secondary, and tertiary impacts; and 3) quantify the climate change mitigation potential of brownfield development in the City of San Francisco.

2. Assessment framework and method

As defined by USEPA, brownfields are "… real property, the expansion, redevelopment, or reuse of which may be complicated by the presence or potential presence of a hazardous substance, pollutant, or contaminant" (USEPA, 2017). Contaminated brownfield sites are often vacant, or under-used as industrial warehouses, storage spaces, etc (Ma et al., 2016). The redevelopment of a contaminated site may result in higher value residential, commercial, industrial, or recreational usage. This conversion in land use is considered to result in an equal number of residential units, commercial units, industrial occupancy, and recreational areas as other potential sites in the region. Quite often, instead of redeveloping a brownfield site, the alternative would be to extend the suburban area, resulting in "urban sprawl". The present study intends to develop an assessment method to quantify the climate change mitigation potential assuming replacing greenfield (i.e., an undeveloped parcel of land) development with brownfield redevelopment.

The ratios of suburban and urban construction compared to total construction, denoted as R_s and R_u respectively, are key parameters in estimating the impact of brownfield redevelopment. A study by Lesage et al. estimated R_s and R_u at 17.8% and 82.2%, respectively, by using an economic partial-equilibrium model (Lesage et al., 2007a). For the purpose of this study, it is assumed that 100% of future development will occur in a suburban area. This assumption was made to quantify the full climate change mitigation potential of brownfield redevelopment.

The proposed assessment method is based on consequential life cycle assessment, which evaluates the environmental impacts associated with changes in which products or services are used (Kua and Lu, 2016; Sevigné-Itoiz et al., 2015). In comparison, the traditional attributional life cycle assessment method is mainly used to determine the environmental impacts associated with making, using, and disposing of a product or service. For contaminated land remediation, Lesage et al. developed a consequential LCA method to assess the full environmental implications on an individual site level for replacing greenfield development with brownfield redevelopment (Lesage et al., 2007b, 2007c). The present study builds upon Lesage et al.'s methods for city-level assessment. Fig. 1 presents a general framework for estimating overall environmental impacts in a consequential LCA for contaminated land redevelopment.

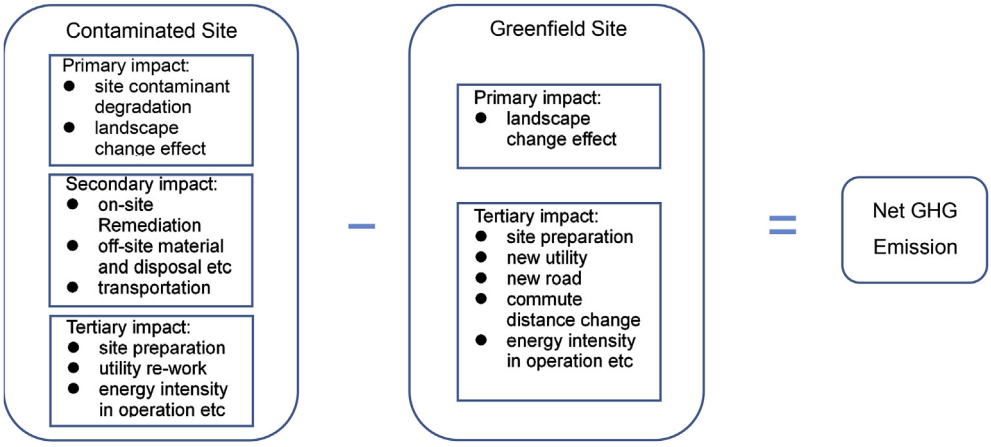

Fig. 1. Illustration of consequential life cycle assessment methodology.

The total climate change mitigation potential is estimated over a pre-determined time period by summing up GHG emission and avoided emission associated with: (1) site contamination and the physical state of the site (e.g., the vegetation); (2) remediation activities; and (3) site usage (see Equation (1) below). These three types of climate change impact are termed primary, secondary, and tertiary impacts (Lesage et al., 2007c).

$$\Delta GHG_{total} = \Delta GHG_p + \Delta GHG_s + \Delta GHG_t \quad (1)$$

where ΔGHG_{total} represents total climate change mitigation potential, in kg of CO_2 equivalent; ΔGHG_p represents the net GHG emissions associated with the degradation of site contaminants (e.g., anaerobic conversion of organic pollutants to methane) and emissions associated with the vegetation of the site (e.g., soil respiration in grassland vs. bare soil); ΔGHG_s represents the cradle-to-grave life cycle GHG emissions associated with all remediation activities, including raw material acquisition and end-of-life waste disposal; and ΔGHG_t represents the GHGs off-set by developing brownfield land rather than greenfield land.

The primary impact can be calculated using the following equation:

$$\Delta GHG_p = \sum_{i=1}^{n}\left(\alpha_i M_i + \beta_i A_i T_p - \chi A_i^r T_p^r\right) \quad (2)$$

where n is the total number of contaminated sites; α_i is the conversion factor for contaminant degradation at the i_{th} site contributing to GHG emissions, in kg/kg; M_i is the total amount of organic contaminants at the i_{th} site, in kg; β_i is the emission factor difference at the i_{th} brownfield site contributing to net GHG emissions due to land use change, in kg/ha-year; A_i is the total area of the i_{th} contaminated site, in hectares (ha); T_p is the duration of primary impact associated with landscape change, in years; A_i^r is total area of the i_{th} greenfield site, in hectares (ha), replacing the i_{th} contaminated site, χ_i is the emission factor difference at the i_{th} greenfield site contributing to net GHG emissions due to land use change, T_p^r is the duration of the primary impact associated with the landscape change at the greenfield site.

The secondary impact can be estimated using existing attributional LCA methods (Hou et al., 2016; Lemming et al., 2010). However, for a city-level assessment, it is not feasible to perform an in-depth analysis for each individual site. An input-output (IO) analysis based LCA would be more suitable for assessing a large number of sites simultaneously. As explained in below, some process data can be collected on an aggregated-level by using existing survey data. Therefore, a hybrid LCA method combining IO and process-based methods is recommended, as explained in existing literature (Hou et al., 2014), to achieve greater accuracy than a pure IO-based LCA and less time expenditure than a pure process-based LCA. Similar hybrid LCA methods have increasingly been used in estimating energy usage and GHG emission (Nagashima et al., 2017; Rocco et al., 2017; Teh et al., 2017).

The tertiary impact can be estimated using the following equation (Hou et al., 2014):

$$\Delta GHG_t = \sum_{i=1}^{n}\left[G_u\left(U_i^b - U_i^g\right) + G_R\left(R_i^b - R_i^g\right) + G_C\left(C_i^b - C_i^g\right)T_C + G_E\left(E_i^b - E_i^g\right)T_E\right] \quad (3)$$

where G_u represents the emission factor for construction of new utilities during redevelopment (ton CO_2 eq per \$); U_i^b and U_i^g represent the construction costs of new utilities (\$) in brownfield redevelopment and greenfield development, respectively; G_R represents the emission factor for construction of new roads during redevelopment (ton CO_2 eq per \$); R_i^b and R_i^g represent the construction costs of new roads (\$) in brownfield redevelopment and greenfield development, respectively; G_C represents the emission factor for the average daily residential commute to/from the redeveloped site to/from work (ton CO_2 eq per km); C_i^b and C_i^g represent the average residential daily commute distance (km/day) to/from work for the brownfield redevelopment and greenfield development, respectively; T_c represents total number of commuting days within the study period; G_E represents the emission factor for energy use by residents living at the site (ton CO_2 eq per unit energy); E_i^b and E_i^g represent average annual energy consumption (energy unit [see Table 3]) by residents of brownfield redevelopments and greenfield developments, respectively; T_E represents the total number of years of residential energy consumption within the study period.

It should be noted that GHG emissions associated with housing construction would also be different for brownfield redevelopment or greenfield development (see Section 3.2.8). With greenfield development, more detached single family houses are likely to be built, resulting in greater construction-related GHG emissions per future resident than compared to construction for brownfield redevelopment. However, due to lack of data, this difference is not evaluated in the present study. Therefore, the climate change mitigation potential estimated here is likely to be somewhat conservative.

In the present study, life cycle GHG emissions were estimated using SimaPro version 8.3.0, with the "IPCC 2013 GWP 100a" method. The life cycle impact assessment results were expressed in tons of CO_2 eq. The recommended time horizon of the GHG inventory is 70 years, which is consistent with the life span used in human health risk assessments (USEPA, 2015). It is assumed that remediation of all brownfield sites could be completed within 20 years, with an average duration of remediation of 10 years; and an average development time of 5 years. Therefore, usage of the developed sites (e.g., household energy consumption) would be 55 years.

3. Case description and data collection

3.1. Case study background

San Francisco is a major city on the western coast of the United States, serving as a cultural, commercial, and financial center. San Francisco has a population of 870,887 as of July 1, 2016 (USCB, 2017), while the larger metropolitan area surrounding San Francisco (i.e., the San Francisco Bay Area) has a population of 7.1 million people as of 2012 (Ekstrom and Moser, 2012). The City of San Francisco is the second-most densely populated major city in the US, with only New York City having a higher population density (Demographia, 2017).

In an era of global climate change, the City of San Francisco is also facing many challenges. Climate simulation models show that heat waves will increase in frequency, magnitude, and duration within the first half of the twenty-first century (Cayan et al., 2012). The Bay-Delta-River system as well as ecological systems in San Francisco are under threat by climate change (Cloern et al., 2011; Galbraith et al., 2002). Due to the impact of climate change on agricultural production, adaptive measures are recommended in order to maintain the viability of certain economic sectors, e.g. the wine industry (Chaplin-Kramer, 2012). Climate change also threatens San Francisco's resilience in water supply, wastewater treatment systems, transportation systems, energy supply, and

public health (Ekstrom and Moser, 2012).

California's Sustainable Communities and Climate Protection Act (SB 375) intends to align future land development with GHG emission goals, incentivizing compact development and vehicle usage reduction, etc. (Darakjian, 2009). Additionally, San Francisco is a global leader in moving toward a low-carbon economy. Its GHG emissions were reduced by 14.5% from 1990 to 2010, whilst the gross domestic product (GDP) increased by 40% and the population increased by 11% during the same period (SFDE, 2013). In 2008, the City of San Francisco passed the Greenhouse Gas Reduction Ordinance, which required that the city reduce GHG emission by 25%, 40%, and 80% below 1990 levels by 2017, 2025, and 2050, respectively (SFPD, 2010).

The population in San Francisco is expected to keep growing (see Fig. 2). It poses a big challenge to accommodate this growing population while reducing GHG emissions. The present study intends to show that brownfield redevelopment can form a significant part of the answer to this challenge. As Fig. 3 shows, historical industrial activities in San Francisco have resulted in a myriad of brownfield sites which are vacant or underdeveloped.

3.2. Data collection and determining key parameters

3.2.1. Number of contaminated sites

The present study obtained contaminated site information from two databases: (1) Geotracker (https://geotracker.waterboards.ca.gov/), which contains records for sites overseen by the California State Water Resources Control Board (SWRCB), and (2) Envirostor (https://www.envirostor.dtsc.ca.gov/), which contains records for sites overseen by the California Department of Toxic Substances Control (DTSC). The SWRCB oversees leaking underground storage tank (LUST) sites and other sites that are likely to primarily impact groundwater. DTSC oversees cleanup, permitting, enforcement, and investigation efforts at hazardous waste facilities and sites with known contamination. Based on a search of these two databases, 235 sites were identified from Geotracker and 187 sites were identified from Envirostor for the city of San Francisco. Because both Geotracker and Envirostor contain records for sites managed by other agencies, some sites are listed in both databases. There are also cases in which one site was assigned more than one ID, and sites that have already been approved for no further action. This study used existing knowledge and professional judgement (e.g., address and business name) to remove replicated sites, resulting in a total of 206 sites.

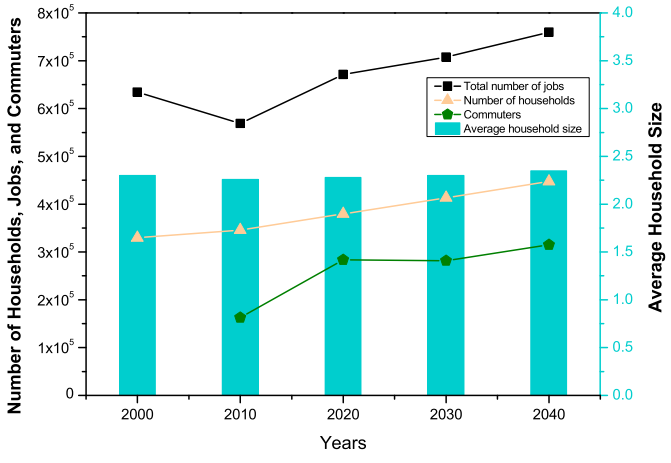

Fig. 2. Growth in number of households, jobs, commuters, and household size (i.e., number of people living in a house) (data source: Census Bureau 2013, Metropolitan Transportation Commission).

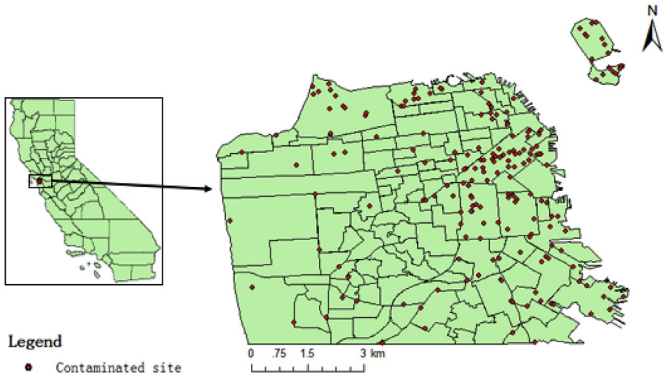

Fig. 3. Map of brownfield sites in the city of San Francisco.

3.2.2. Areal differential

For a successful urban brownfield project redevelopment project, a significantly smaller amount of land area is required than for the equivalent development of a greenfield site. This effect is described by the "areal differential" (AD) factor, which equates to the acreage of greenfield that would have been used, divided by the acreage of the brownfield that is used. Sherk (2002) derived ADs of 6.2, 2.4, and 5.4 for industrial, commercial, and residential use, respectively, in the US (Sherk, 2002). Based on data from the San Francisco Planning Department (SFPD, 2015), it was estimated that future development of vacant and underdeveloped sites would result in 53% industrial, 28% commercial, and 18% residential use of the developed areas (see Fig. 4). Based on this distribution of land use, the weighted average AD is estimated to be 4.7.

3.2.3. Urban and suburban/remote sites

While most existing studies of brownfield redevelopment have focused on urban sites, the present study has included all contaminated sites, including those located in suburban areas (e.g., remote military bases). As discussed in Section 3.2.2, development in urban areas has a significantly different GHG emission footprint compared to suburban areas. Therefore, this study have classified contaminated sites into two categories: urban sites and suburban/remote sites. For the former type, AD is taken into consideration; while for the later type, it is assumed that the AD is 1. In a typical city where the downtown center is located in the geographic center of the city, it can be assumed that urban sites are located within half of the 'diameter' of the city limits. San Francisco is unique in that its

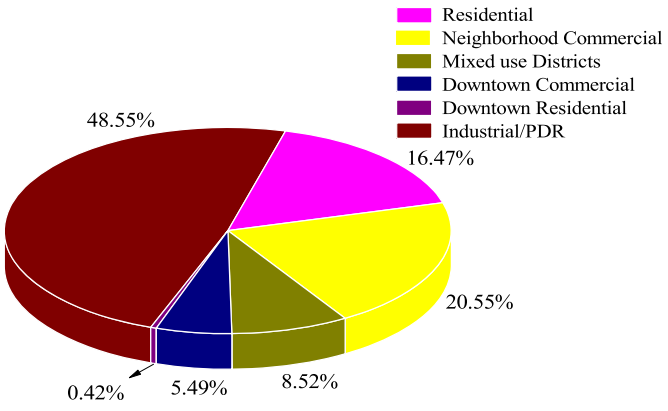

Fig. 4. Expected land use distribution of new development in San Francisco.

downtown center is located in the northeast corner of the city due to historical reasons. Therefore, this study calculated the 'radius' of the city as the distance from the downtown center to the furthest point of the city boundary, and counted any sites with half the radius as urban. This resulted in 168 urban sites and 38 suburban/remote sites.

3.2.4. Average contaminated site size

The size of each contaminated site partially determines its redevelopment value and corresponding climate change mitigation potential. Therefore, it is a critical parameter. However, Geotracker and Envirostor do not include site acreage for all contaminated sites. To fill this information gap, this study estimated an average size for all sites listed in Geotracker and Envirostor. A total of 68,287 sites were screened, and site size information was obtained for 5685 sites (Table 1). Based on these data, it was estimated that California has nearly 2.3 million ha of potentially contaminated land, representing 5.5% of California's total land area. San Francisco was estimated to have nearly two thousand ha of potentially contaminated land, representing 7.6% of its land area. As can be seen in Table 1, Superfund sites and military sites tend to be large, but the standard error is also high. This is because some of these facilities tend to be very large facilities while many others are relatively small.

3.2.5. Effect of landscape change

Urban areas represent a main source of GHG due to anthropogenic activities. On the other hand, urban areas can also consist of a significant amount of vegetation, which serve as a carbon sink due to photosynthesis (Hiller et al., 2011). Qian and Follett reported an average carbon sequestration rate of 0.9–1.0 ton ha^{-1} yr^{-1} for urban areas, based on an analysis of 45 years of data (Qian and Follett, 2002). Under natural conditions, urban turf can be a significant carbon sink. However, the increased usage of fertilizer could change lawns from a carbon sink to carbon source, due to CO_2 emissions associated with fuel combustion, fertilizer production, and irrigation (Townsend-Small and Czimczik, 2010). For the present study, it is assumed that 45% of the total development area will be occupied by urban turf, based on US data (Milesi et al., 2005). The emission factors associated with the landscape at developed and greenfield sites are estimated to be 5.8 ton CO_2 eq ha^{-1} yr^{-1} and -1.9 ton CO_2 eq ha^{-1} yr^{-1}, respectively, based on studies conducted in California (Townsend-Small and Czimczik, 2010; Xu and Baldocchi, 2004). Consequently, the emission factors for brownfield redevelopment and greenfield development were estimated to be 4.6 ton CO_2 eq ha^{-1} yr^{-1} and 7.6 ton CO_2 eq ha^{-1} yr^{-1}, respectively.

3.2.6. Average remediation effort

Due to the large number of contaminated sites within the study area, it is not feasible to conduct an in-depth analysis for each site. Therefore, a simplified method is required to estimate the "average" remediation effort. As discussed above (Section 2), this study recommends the use of a hybrid LCA method for estimating secondary impacts. Based on data from the literature, this study estimated the total cost to remediate all sites in San Francisco at $1.742 billion in 2002 dollars (see Table S1). The $0.679 billion costs for remediation via excavation and off-site disposal were estimated using a process-based approach. According to USEPA, excavation and off-site disposal accounted for 43% of RCRA corrective actions (USEPA, 2004); and that among 459 decision documents for Superfund sites, 24% used excavation and off-site disposal (USEPA, 2013). Other techniques would be used for the remaining remediation, with an estimated cost of $1.063 billion. Other frequently used remediation techniques include on-site containment, soil vapor extraction, stabilization/solidification, chemical oxidation/reduction, thermal treatment, etc. (USEPA, 2013). The remaining costs were allocated among the various remediation technologies based upon their adoption rate (see Table S4).

Remediation time is also a critical parameter in determining the tertiary impact. Some active remedies (e.g., dig and haul) last from several months to a few years; while some passive remedies (e.g., monitored natural attenuation [MNA]) may takes decades to complete. In the present study, it is assumed that active remediation is combined with sustainable means that do not affect land redevelopment (e.g., vapor barriers and MNA), resulting in an average remediation duration of 10 years. A sensitivity analysis was conducted on this parameter (see Section 3.3). More details about remediation efforts are included in the Supplementary Materials.

3.2.7. Housing densities

In order to estimate tertiary impacts, it is important to calculate the expected number of housing units that will be built at each contaminated site. Based on San Francisco's generalized permitted housing densities by zoning districts, housing density increases with proximity to the downtown area. Using the housing density map (SFPD, 2015), the average housing density was estimated for sites with various distances from the downtown center, corresponding to low, moderately low, medium, moderately high, and high densities (see Table 2). It should be noted that San Francisco is unique in that it has three sides facing water: either the Pacific Ocean or the San Francisco Bay. The water front has high development value. Consequently, many areas that are far away from the downtown center still have a medium level housing density. In the present study, based on the housing density map, it was assumed that contaminated sites within 1 km of the water front would have

Table 1
Contaminated site number and size estimation for the State of California.

	Average Area (ha)	Standard Error (ha)	California		San Francisco	
			Number of Sites	Total Area (ha)	Number of Sites	Total Area (ha)
Superfund Sites	529	89	110	58,213	6[a]	170
Military Sites	445	51	2033	904,603	21	107
Corrective Action Sites	22	3.7	602	13,009	5	108
Land Disposal Sites	2.6	0.5	3409	8886	8	21
Cleanup Sites	15	2.1	12,150	187,987	27	160
School Sites	6.2	0.2	1668	10,402	2	12
Evaluation Sites	4.7	0.6	1769	8328	32	110
Historical Sites	4.3	1.8	2194	9365	15	64
State Response Sites	14	1.9	723	10,166	12	27
UST Sites	1.9	0.4	43,599	81,197	78	145
			TOTAL	*2,349,094*		*925*

[a] There is only one Superfund site, but it was divided into six parcels in the Envirostor system.

Table 2
Estimated housing densities in various city areas.

	Distance from downtown center (km)	Housing density (houses per ha)
Low density	>4.5	35
Moderately low density	3–4.5	89
Medium density	2–3 (or within 1 km of water front)	133
Moderately high density	1–2	225
High density	1	699

Note: the relationship between housing density and distance from downtown center represent an approximation of average conditions, rather than a strict boundary condition; data source: (SFPD, 2015).

at least a medium housing density, regardless of its distance from the downtown center.

3.2.8. Housing types and energy intensity

Housing types in urban developments differ from those in suburban areas. More single family detached houses are expected for suburban areas and more multi-unit houses are expected for urban areas. In North America, single family homes are a typical stand-alone dwelling type where one family live, which is often more spacious and requires more space heating energy. Multi-unit family homes are a dwelling type which contain multiple separate housing units within one building, being inhabited by multiple families. Lesage et al. (2007c) used construction statistics to estimate the dwelling type: family residence was predominantly attached dwellings in urban areas (50%), and predominantly detached dwellings in suburban areas (80%) (Lesage et al., 2007b). In this study, the housing type for brownfield developments was determined based on its location. Table 3 below shows the housing type distribution for different areas, derived using data from San Francisco's Planning Department (SFPD, 2015). In calculating "avoided" emissions, a "low density" development is used to estimate the housing type at a "replaced" greenfield development site. Household energy consumption data were obtained from the Residential Energy Consumption Survey (RECS; USEIA, 2013). RECS is conducted every 4–6 years, with the last two surveys being conducted in 2009 and 2015, respectively. However, the 2015 consumption data will not be made available until 2018, therefore, data from the 2009 survey was used (see Table 4).

3.2.9. Residential commute

The commute to work distance is expected to be lower for residents of urban developments than suburban developments, as all residents are assumed to work in urban areas. According to SFPD (2015), 27.3% of employees in the City of San Francisco commuted from outside of the city, which is projected to increase to 43% in 2020 (SFPD, 2015). The city aims to reduce the number of out-of-city commuters, which can be partly fulfilled by redeveloping brownfields. For the purpose of this study it was assumed that the average commute distance (two ways) for greenfield development residents would be 51 km (Mokhtarian et al., 2004), whereas the commute distance for brownfield development residents was measured as the distance from downtown San Francisco to each brownfield site. According to the American Community Survey, 70% of commuters in the San Francisco metropolitan commute using automobiles (McKenzie, 2015). Thus, redeveloping brownfield land was assumed to result in GHG savings by reducing the daily commute of residents. While there will some additional savings due to commuters that may switch to using either public transportation or cycling following brownfield redevelopment, this was assumed to represent a relatively small percentage of commuters, resulting in a minor savings. Therefore, cost savings from the use of public transport and cycling are not evaluated here. According to data and projections from SFPD (2015), the number of workers per household will grow from 1.22 in 2010 to 1.28 in 2030 in the San Francisco area. Based on the overall trend, the worker to household ratio of 1.28 was used in the present study (SFPD, 2015). The average automobile emission factor was obtained from the US Department of Transportation (USDOT, 2010).

Table 4
California household energy consumption by house type[a].

Energy Type	Usage Type	Single-Family	Multi-Family
Electricity (kWh)	Total	7881	4007
	Space Heating	991	994
	Water Heating	2089	1324
	Air Conditioning	1496	658
	Refrigerators	1208	722
	Other	5372	2192
Natural Gas (thousand cubic feet)	Total	46	20
	Space Heating	24	13
	Water Heating	19	10
	Other	12	7
Propane/LPG (gallons)	Total	420	294
	Space Heating	340	340[b]
	Water Heating	205	137
	Other	112	112[b]
Fuel Oil (gallons)	Total	361[b]	361[b]
Kerosene (gallons)	Total	48[b]	48[b]

Note:
[a] Derived using data from (USEIA, 2013).
[b] Data not available and assumed to be equal to that of the overall value for Western US.

3.2.10. Utility

Public utilities, including potable water, sewer, electricity, and gas lines are required for new developments, regardless of whether the development is a brownfield or a greenfield. The length of these utilities at a brownfield is assumed to be the same as the average length of all utilities in an urban setting. To estimate the total length of utilities in suburban settings, the average length of all utilities in an urban parcel is multiplied by $\sqrt{AD}$. GHG emissions associated with utility maintenance is not evaluated here. For greenfield development, utility lines are expected to be longer as they must also link the new site with existing hubs, but this effect is ignored

Table 3
Distribution of housing structure type.

	Percent of single family home	Percent of multi-unit family home	Representative district
Low density	80%	20%	South Central
Moderately low density	39%	61%	Central
Medium density	10%	90%	South Market
Moderately high density	5%	95%	Northeast
High density	2%	98%	Downtown

Note: the relationship between housing density and distance from downtown center represent an approximation of average conditions, rather than a strict boundary condition; data source: (SFPD, 2015).

due to a lack of data. Due to the above two omissions, the estimate in the present study is considered to underestimate of the benefit of brownfield redevelopment. There is insufficient data to conduct a process-based inventory for utility construction on a city scale. Thus, in order to achieve a reasonable estimate of the environmental impact associated with utility construction, this study used a combination of costing and IO analysis. For water and sewer lines, costing was conducted using the service charged by the San Francisco Public Utilities Commission (SFPUC) (SFPUC, 2016). Gas and utility connection costs were estimated using the service charged by Pacific Gas and Electric (PG&E), the gas and electric utility in the San Francisco area. More details are provided in the Supplementary Materials.

3.2.11. Road

Urban sprawl results in an expansion of roadway networks. It has been shown in Australia, that capital costs associated with road construction for urban fringe developments were 6 times that of inner city developments (Trubka et al., 2008). In the present study, it is assumed for greenfield development that new lane construction equal to the perimeter of the development area would be required. According to the Transportation Cost and Benefit Analysis by the Victoria Transport Policy Institute, right of way for new lanes cost $209,000 (in 2000 dollars) per lane-mile for outlying areas, and construction of new lanes cost $1,728,000 per lane mile (Litman, 2009). The increased cost of road infrastructure due to increased capacity demand is assumed to be 1/6 of the capital cost for road construction in greenfield development. It was assumed that the nearby roadways have two lanes in each direction.

3.3. Sensitivity analysis

Sensitivity analyses were conducted as follows: (1) remediation durations of 5 and 20 years were evaluated in addition to the base case of 10 years (Section 3.2.6); (2) the AD was changed from a weighted average of 4.7 (i.e., the base case in Section 3.2.2) to a value of 5.4, which assumes that all redevelopment is residential; (3) the organic contaminant mass was doubled; (4) landscape emissions were doubled from the base case assumed in Section 3.2.5; (5) remediation efforts were adjusted by doubling the dig and haul effort or by doubling other remedial efforts from the base case assumed in Section 3.2.6; (6) housing density was increased or decreased by 30% from the base case assumed in Section 3.2.7; (7) the percentage of single family dwellings in suburban areas was reduced by 50% from the base case assumed in Section 3.2.8; (8) an extra 30% energy savings was assumed for single family dwellings from the base assumed in Section 3.2.8; (9) automobile GHG emissions were reduced by 30% from the emissions factors used in Section 3.2.9, and (10) utility or road density was increased by 30% from the base assumed in Section 3.2.10.

4. Results

4.1. Overall life cycle impacts

The calculated changes in life cycle GHG emissions due to the remediation and redevelopment of contaminated land as compared to greenfield land in San Francisco are shown in Fig. 5.

Overall, the redevelopment of brownfield sites replacing the development of greenfield sites resulted in a net GHG emissions reduction of 51.9 Mt CO_2 eq. over 70 years, or 0.74 Mt CO_2 eq. yr^{-1}. This can be divided as follows: The primary impact (i.e., associated with the site contamination and the physical state of the site) rendered a net GHG emissions reduction of 1.2 Mt CO_2 eq. over 70 years; The secondary impact (i.e., associated with remediation activities), resulted in a net GHG emission increase of 3.5 Mt CO_2 eq. over 70 years; and, the tertiary impact (i.e., associated with the consequential change in land use and household energy consumption) led to a net GHG emissions reduction of 54.2 Mt CO_2 eq. over 70 years. Integrating USEPA's social cost of carbon of $56 (2007 USD) per metric ton of CO_2 at a 2.5-percent discount rate (USG 2016), an approximate reduction of $3.39 billion (2017 USD) of global monetized impacts due to climate change effects is achieved (Harclerode et al., 2016).

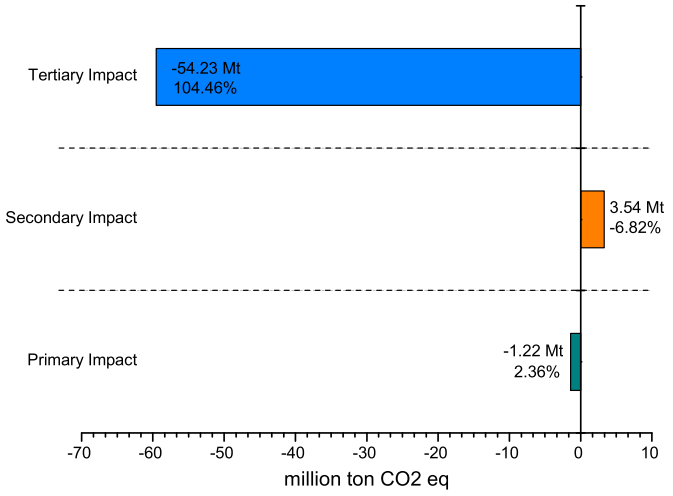

Fig. 5. Life cycle GHG emission and reduction associated with contaminated land remediation and redevelopment in San Francisco.

To put these numbers in perspective, the GHG emissions from within the limits of the city of San Francisco totaled 5.3 Mt CO_2 eq. in the year of 2010 (SFDE, 2013). Our results show that brownfield redevelopment can potentially result in a 14% reduction in GHG emission for the entire city compared to the 2010 level. This represents a significant climate change mitigation potential. Among the various climate change mitigation measures the City have proposed, only “moving to 100% renewable electricity” would have a higher GHG emission reduction potential (i.e., 0.94 Mt CO_2 eq) (SFDE, 2013).

4.2. Geographic distribution

Fig. 6 shows the spatial distribution of contaminated sites and their associated climate change mitigation potential. As can be seen, contaminated sites with high climate change mitigation potential are mostly located in the eastern side of the city, especially the areas near the downtown center. This is understandable because development in this area is expected to be mostly high-rise buildings, for which household energy consumption would be lower and commute distances would also be substantially lower. Another reason is that this side of the city was more heavily developed due to historical reasons and, therefore, contained more contaminated sites. Most notable are several sites in the southeastern corner of the city with a high climate change mitigation potential. These locations represent several parcels in the Hunters Point Naval Shipyard Superfund site. This area has a very high climate change mitigation potential because it represents a large vacant area suitable for moderate to high density development. The Hunters Point Shipyard and Candlestick Point is currently undergoing redevelopment activities transferring contaminated land into residential units, including affordable housing.

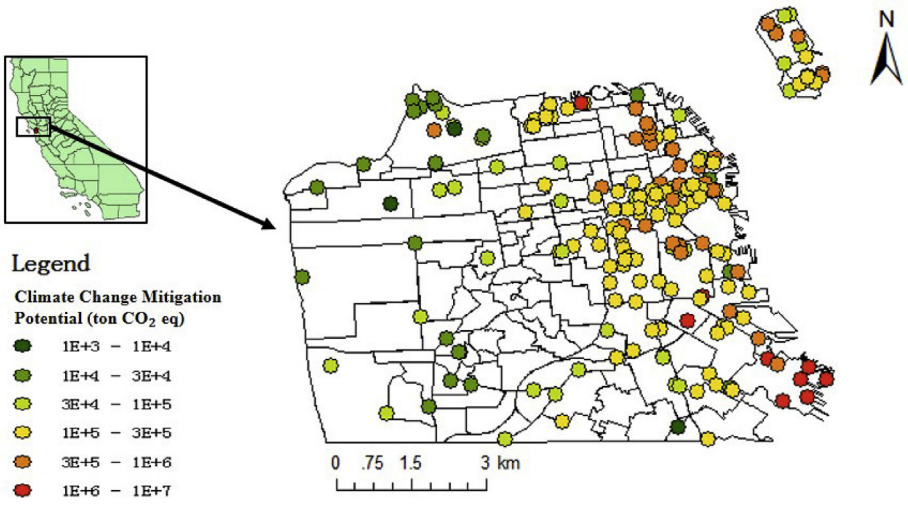

Fig. 6. Spatial distribution of GHG reduction at contaminated sites due to brownfield redevelopment.

4.3. Contribution analysis

A contribution analysis was conducted to identify the critical processes in the overall GHG emission and reduction. As Fig. 7 shows, for primary impacts, the change of landscape at greenfield sites, i.e. converting a net carbon sink (Xu and Baldocchi, 2004) into a net carbon source (Townsend-Small and Czimczik, 2010), resulted in the biggest contribution in net GHG emissions. Whereas brownfield redevelopment avoids this impact. The transformation of organic contaminants during remediation resulted in a significant amount of GHG emissions (−18.9%), as did landscape change at brownfield sites (−19.1%); however, both emissions were offset by the much higher GHG emissions reduction at greenfield sites. For secondary impacts, numerous processes could contribute to GHG emissions because many different remedial technologies are used. The present study found that waste disposal accounted for a significant portion of GHG emissions (38.2%). "Other processes" in Fig. 7 represent a large group of remedial processes, whose GHG emissions were not individually identified; instead, GHG emissions were estimated using an IO method (59.2%). For tertiary impacts, the change in household energy consumption contributed to the largest GHG emissions reduction (41.2%), including electricity reduction (20.2%), natural gas reduction (13.6%), and propane/LPG reduction (7.4%). The second largest GHG emissions reduction was a reduction in commute distance (34.0%). The reduction in utility infrastructure construction (including water lines, sewer lines, gas lines, and electricity lines), and reduction in road construction contributed to GHG emissions reductions of 21.3% and 3.5%, respectively.

4.4. Sensitivity analysis

As can be seen in Fig. 8, the sensitivity analysis suggested that most parameters had relatively little effect on the overall results. The results were not very sensitive to many of the parameters changed, including organic contaminant mass, brownfield and greenfield emission factors, remediation efforts, and road densities. This is because these processes had a relatively small contribution to the overall climate change mitigation potential. However, the results were sensitive to several key parameters, including remediation time, housing density, the percentage of single family houses in a suburban area, and energy savings by single-family

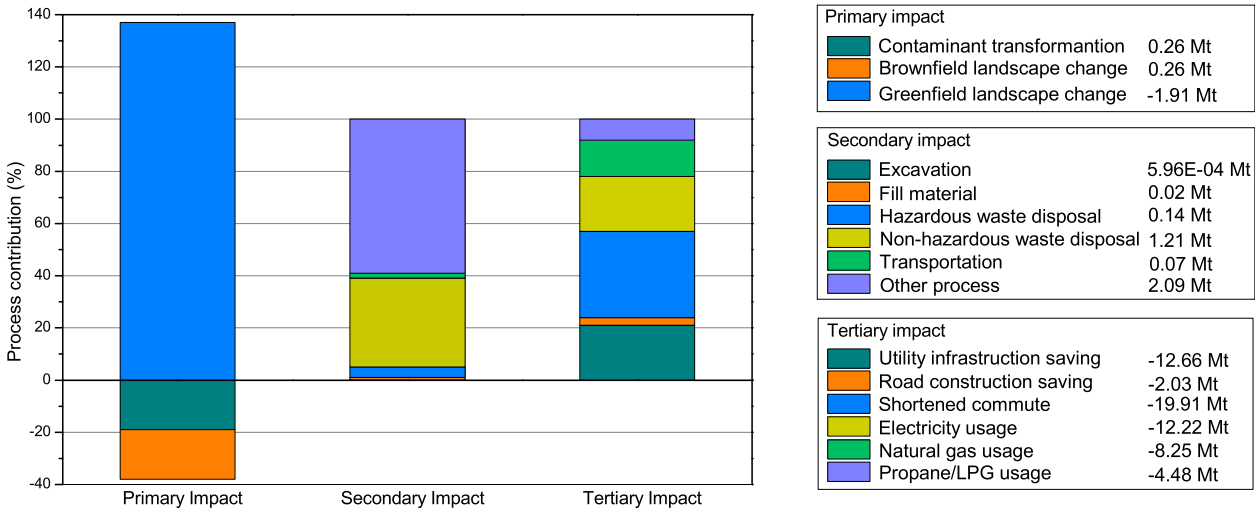

Fig. 7. Percent contribution of different components contributing to life cycle GHG emission and reduction.

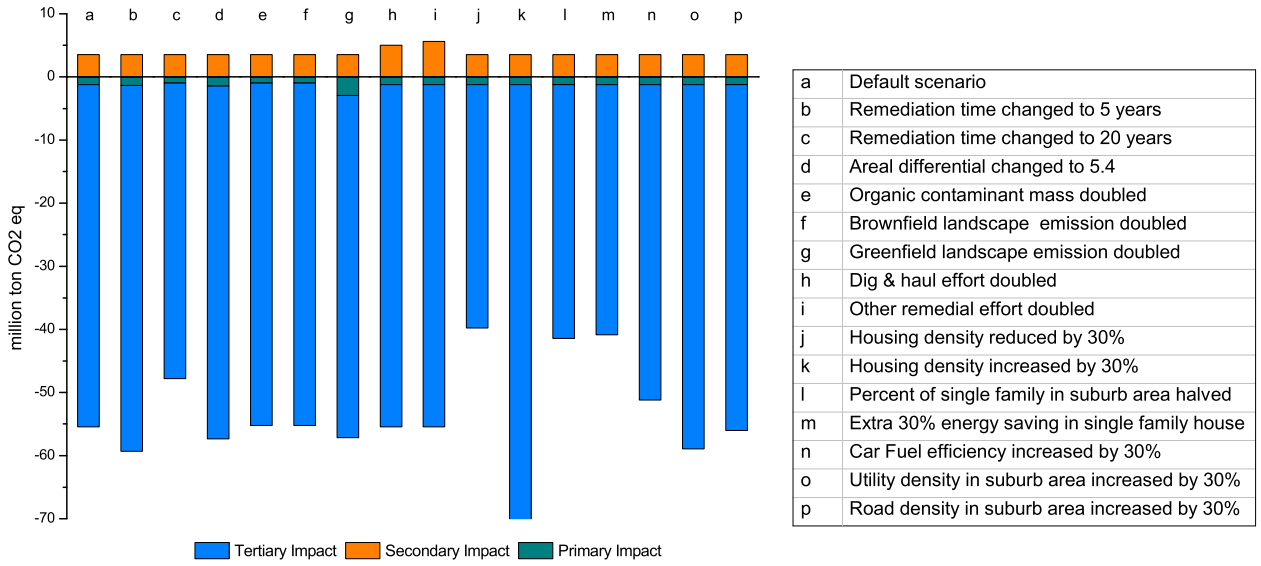
a b c d e f g h i j k l m n o p
10
0
-10
-20
-30
-40
-50
-60
-70
million ton CO2 eq
Tertiary Impact
Secondary Impact
Primary Impact
a Default scenario
b Remediation time changed to 5 years
c Remediation time changed to 20 years
d Areal differential changed to 5.4
e Organic contaminant mass doubled
f Brownfield landscape emission doubled
g Greenfield landscape emission doubled
h Dig & haul effort doubled
i Other remedial effort doubled
j Housing density reduced by 30%
k Housing density increased by 30%
l Percent of single family in suburb area halved
m Extra 30% energy saving in single family house
n Car Fuel efficiency increased by 30%
o Utility density in suburb area increased by 30%
p Road density in suburb area increased by 30%

Fig. 8. Sensitivity analysis of critical parameters: a shows the default scenario, b through p show the effect of changing key parameter values.

houses. When remediation time increased from 10 years to 20 years, the GHG emissions reduction decreased from 51.9 Mt CO_2 eq to 44.3 Mt CO_2 eq, representing a 15% drop. The applied changes in housing density brought about the largest impacts of all parameters. When housing density decreased by 30%, GHG emissions reduction decreased to 36.2 Mt CO_2 eq.; and when housing density increased by 30%, GHG emissions reduction increased to 67.6 Mt CO_2 eq. Other significant parameters included when the percentage of single family dwellings in suburban areas was halved, or the energy consumption of single family dwelling was decreased by 30%, for which the GHG reduction emissions decreased to 37.9 and 37.3 Mt CO_2 eq, respectively.

5. Discussion

The redevelopment of brownfield land is viewed as a 'smart growth' strategy. It can restrict low-density development which frequently sprawls out from the edge of urban areas, resulting in greenfield occupation and environmental deterioration (Greenberg et al., 2001). This study presents an LCA based approach for estimating the climate change mitigation potential of brownfield redevelopment on a city level, by considering primary, secondary, and tertiary impacts. While LCA has commonly been used to evaluate the environmental benefits and costs associated with remediation, most remediation LCA studies have focused on secondary impacts, only a few of the more recent studies have evaluated both primary and secondary impacts (Hou et al., 2015, 2016; Lemming et al., 2010), and there are fewer still studies evaluating tertiary impacts (Hou et al., 2014; Lesage et al., 2007c). According to Owsianiak et al., among 31 typical remediation LCA studies published in peer-reviewed literature, 17 did not include primary impacts and 9 used inappropriate models to quantify primary impacts (Owsianiak et al., 2013). The lack of studies on tertiary impacts is likely due to the large amount of data required, as well unfamiliarity among those with access to the necessary data (i.e., urban planners and researchers conducting city-level assessments). However, as demonstrated here, tertiary impacts can have a much greater magnitude than primary and secondary impacts; and therefore, should be considered in environmental management decision making.

The method presented here is for city-level assessments, in which average parameters are used. These parameters should not be used as default values in a site-specific assessment, because the context is different. For instance, when estimating tertiary impacts for an individual site, one needs to take into account spill-over effects. The most probable scenario should be based on a partial equilibrium model (Lesage et al., 2007c). Lesage estimated that a residential development on a one acre brownfield site would replace 0.82 acres of other urban redevelopment, increasing the total developed urban area by 0.18 acres (Lesage et al., 2007b). In comparison, when this study estimated tertiary impacts at a city level, it should not be based on the concept of equilibrium between supply, demand, and usage of price elasticity. Rather, a more straightforward assumption would be to estimate the benefit from brownfield redevelopment by comparing the GHG emissions reduction vs. developing suburban greenfield, which better reflects the aggregated potential benefit of redevelopment of a large number of contaminated sites.

It should be noted that, while this study assumed a certain distribution of residential versus industrial/commercial land use based on San Francisco Planning Department's data, the tertiary impacts estimated here are mainly the product of residential energy consumption and commuting. This is considered appropriate because the consequential impacts associated with industrial and commercial usage would be too complex to model and are expected to be far less significant than the difference in residential energy usage between greenfield and brownfield developments. Several other assumptions, as discussed in Section 3.2, would also result in an underestimate of the climate change mitigation potential of brownfield redevelopment. Therefore, this study assumed the results presented here are conservative and an underestimate of the climate change mitigation potential of brownfield redevelopment.

The present study assumed an average of 10 years to complete remediation at all of the sites. This is by no means a realistic estimate; instead, it is an assumption used to calculate the "potential" would that governments implement effective measures and policies to promote brownfield redevelopment. In reality, the investigation and cleanup of many contaminated sites (especially large sites) in California currently takes 20–30 years, due to cumbersome government requirements, slow approval processes, community/stakeholder lawsuits, and conflicts between feasibility constraints and unrealistically stringent cleanup levels (Hou, 2017; NRC, 2012;

Stroo and Ward, 2010; Sun et al., 2016). One may argue that cleanup targets are mandatory and cleanup should take as long as it currently takes in order to protect health and the environment; however, experience elsewhere has proved otherwise. The London Olympic Park project, for example, cleaned-up a large complex site in less than 3 years (Hou et al., 2015). Furthermore, the UK government enforced a public policy mandating that new housing is preferentially built on brownfield over greenfield land. Consequently, in 2008, 79% of all new dwellings in England were built on previously developed land (DCLG, 2009). We argue that the cleanup of contaminated land could and should be accelerated. The newly emerged green and sustainable remediation movement provides an incentive and a tool to achieve this goal (Hou and Al-Tabbaa, 2014).

Therefore, it is important for policy makers to accelerate remediation to prepare brownfield sites for redevelopment. Strategies include financial and policy instruments to incentivize or even mandate a certain portion of brownfield redevelopment, as was seen in the UK. Liability clauses to mitigate developer's concerns, better inter-governmental cooperation to accelerate decision making and approval, more flexible mechanisms for setting realistic remedial objectives and planning for development accordingly, should also be given consideration. This is also of implications to emerging remediation markets, like China, where a new Soil Action Plan is expected to stimulate hundreds of billions of dollars in remediation spending (Hou and Li, 2017; Zhang and Li, 2016). Secondly, the higher housing density of brownfield sites significantly increases their climate change mitigation potential. However, it should be noted that this would increase the pressure on a number of city services and infrastructure. Thus, there will in practice be an upper limit to housing density. Thirdly, newly developed technologies may enhance the benefit of remediation. For instance, recently researchers have looked into the use of biochar and other biological waste based materials in contaminated land remediation (Li et al., 2016; Shen et al., 2016, 2017b), which has great climate change mitigation potential (Hou et al., 2016).

6. Conclusions

The present study proposed a methodology to quantify the city-wide benefit of brownfield redevelopment in combating global climate change, and used San Francisco as a case study to demonstrate the method. The results showed that brownfield redevelopment offers substantial climate change mitigation potential, equivalent to 14% of the city's annual GHG emissions annually. It confirmed our hypothesis that such city-level assessments are feasible by using aggregated data and simplified assumptions. Sensitivity analysis suggested that most parameters had relatively little effect on the overall results, other than the time it takes to complete remediation. It also showed that land use parameters (such as housing density) affected the final results. While there are a number of limitations and uncertainties associated with the present study, our assessment confirmed that the climate change mitigation potential of brownfield redevelopment is substantial with a high degree of confidence. Therefore, it is suggested that more realistic and city-specific assumptions and parameter values be developed to derive more accurate estimates. This will empower regulators and policy makers to better align their climate change mitigation strategies with other urban planning and development actions. There are a variety of obstacles in cleaning-up contaminated land for redevelopment; however, with political will and government inter-departmental collaboration, brownfield cleanup and redevelopment can be accelerated and contribute to climate change mitigation.

Acknowledgement:

This work was partly supported by the National Natural Science Foundation of China (Grant No. 41671316), the Thousand Talents Program of the Chinese government and Tsinghua University.

Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.jclepro.2017.10.071.

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